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Environment International 30 (2004) 721–739
ENVIRONMENT INTERNATIONAL
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Review article

Life cycle assessment Part 2: Current impact assessment practice

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Received 10 February 2003; accepted 17 December 2003

Abstract

Providing our society with goods and services contributes to a wide range of environmental impacts. Waste generation, emissions and the consumption of resources occur at many stages in a product's life cycle—from raw material extraction, energy acquisition, production and manufacturing, use, reuse, recycling, through to ultimate disposal. These all contribute to impacts such as climate change, stratospheric ozone depletion, photooxidant formation (smog), eutrophication, acidification, toxicological stress on human health and ecosystems, the depletion of resources and noise—among others. The need exists to address these product-related contributions more holistically and in an integrated manner, providing complimentary insights to those of regulatory/process-oriented methodologies. A previous article (Part 1, Rebitzer et al., 2004) outlined how to define and model a product's life cycle in current practice, as well as the methods and tools that are available for compiling the associated waste, emissions and resource consumption data into a life cycle inventory. This article highlights how practitioners and researchers from many domains have come together to provide indicators for the different impacts attributable to products in the life cycle impact assessment (LCIA) phase of life cycle assessment (LCA).

Keywords: Life cycle assessment; Waster generation; Emission

1. Introduction

The methodology for conducting a life cycle assessment (LCA) of products (goods and/or services) consists of the four phases in Fig. 1 (ISO 14040, 1997). In a previous article, Rebitzer et al. (2004) described the first two phases:

1. how practitioners define the goal and the scope of an LCA study and the functional unit[1], and
2. how they tabulate an inventory of the wastes generated, the emissions, as well as the resources consumed per functional unit at each stage in the product's life cycle (that is, from its cradle to its grave—raw material extraction through to ultimate disposal).

This paper describes the life cycle impact assessment (LCIA) phase, focusing on the key attributes of the supporting models and methodologies. These models and methodologies provide LCA practitioners with the factors they need for calculating and cross-comparing indicators of the potential impact contributions associated with the wastes, the emissions and the resources consumed that are attributable to the provision of the product in a study.

LCIA consists of both mandatory and optional elements, as illustrated in Fig. 2 (ISO 14042, 2000):

- Selection of the impact categories of interest, the indicators for each impact category and, although often implicitly considered by practitioners, the underlying models (a procedure also considered in the initial goal and scope phase of an LCA).
- Assignment of the inventory data to the chosen impact categories (*classification*).

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[1] As described in Part 1 (Rebitzer et al., 2004): For example, alternative types of packaging are comparable on the basis of 1 m^3 of packed and delivered product—the service that the product provides and not necessarily the mass of a packaging material.

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doi:10.1016/j.envint.2003.12.009

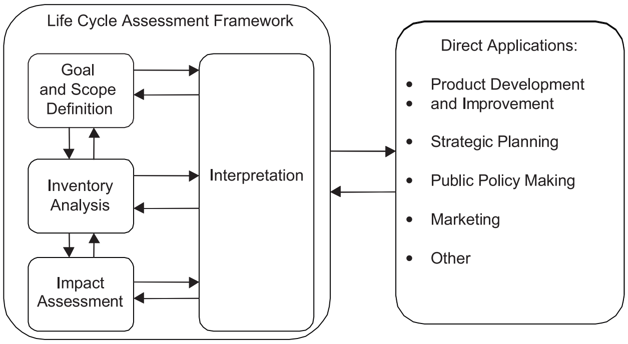

Fig. 1. Phases and applications of a LCA (ISO 14040, 1997).

- Calculation of impact category indicators using characterisation factors (*characterisation*).
- Calculation of category indicator results relative to reference values(s) (*normalisation*, optional).
- *Grouping* and/or *weighting* the results (optional, weighting not being allowed when following ISO14042 in comparative assertions disclosed to the public).
- *Data quality analysis* (mandatory in comparative assertions disclosed to the public, according to ISO 14042, but receiving little attention in current practice).

The next sections outline the key issues of these LCIA elements—starting with an overview of classification and characterisation, a discussion of key modelling issues, and then outlining differences compared to other common risk/impact assessment approaches. Section 6 summarises the models and associated indicators that currently exist for characterisation for commonly adopted impact categories. Given all the indicators for the different impact categories, Section 7 outlines how indicator results can be compared, or condensed, across impact categories using social science techniques when direct comparisons using natural science are not feasible or are considered undesirable.

2. Impact categories and areas of protection (AoPs)

According to ISO 14042, the LCIA standard, there are three broad groups of impact categories that should be taken into account when defining the scope of an LCA study. Impact categories include climate change, stratospheric ozone depletion, photooxidant formation (smog), eutrophication, acidification, water use, noise, etc. The three broad groups are commonly referred to as AoPs (Udo de Haes et al., 1999):

- Resource use
- Human health consequences
- Ecological consequences

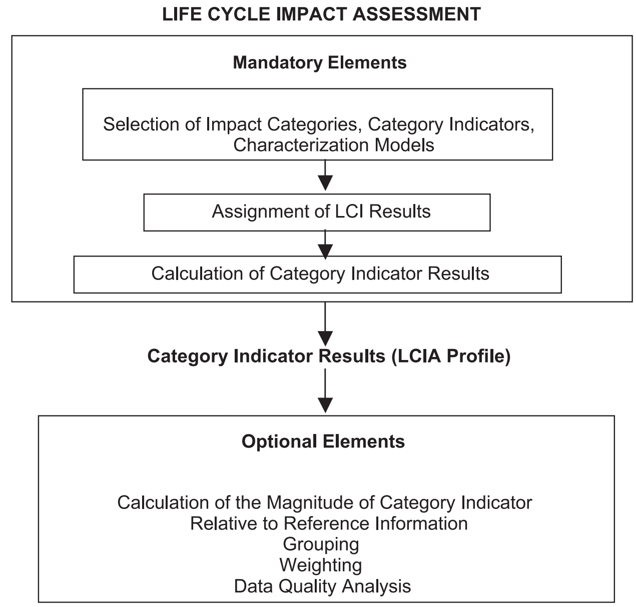

Fig. 2. Elements of LCIA (ISO 14042, 2000).

In some of the recent proposals (e.g. Udo de Haes and Lindeijer, 2002), these AoPs have been re-organised as in Fig. 3. The suggested AoPs are now:

- human health
- natural environment (resources and life support functions—climate regulation, soil fertility)
- man-made environment (monuments, forest plantations)

Fig. 4 illustrates for the Japanese method the overall links between the inventory data, the category indicators, the areas of protection, as well as the optional step of weighting across these AoPs to provide a final impact indicator for a product, in this case in the monetary terms of Yen.

3. Characterisation

Eq. (1) provides an example for emissions data of how indicators for each impact category can be readily calculated from the inventory data of a product using generic *characterisation factors*. These factors are typically the output of *characterisation models*. The factors are made available to practitioners in the literature, in the form of databases, as well as in available LCA support tools. Similar generic equations and data exist for wastes and resource consumption, as outlined further in Section 6.

$$\text{Category Indicator} = \sum_s \text{Characterisation Factor}(s) \times \text{Emission Inventory}(s) \qquad (1)$$

where subscript s denotes the chemical.

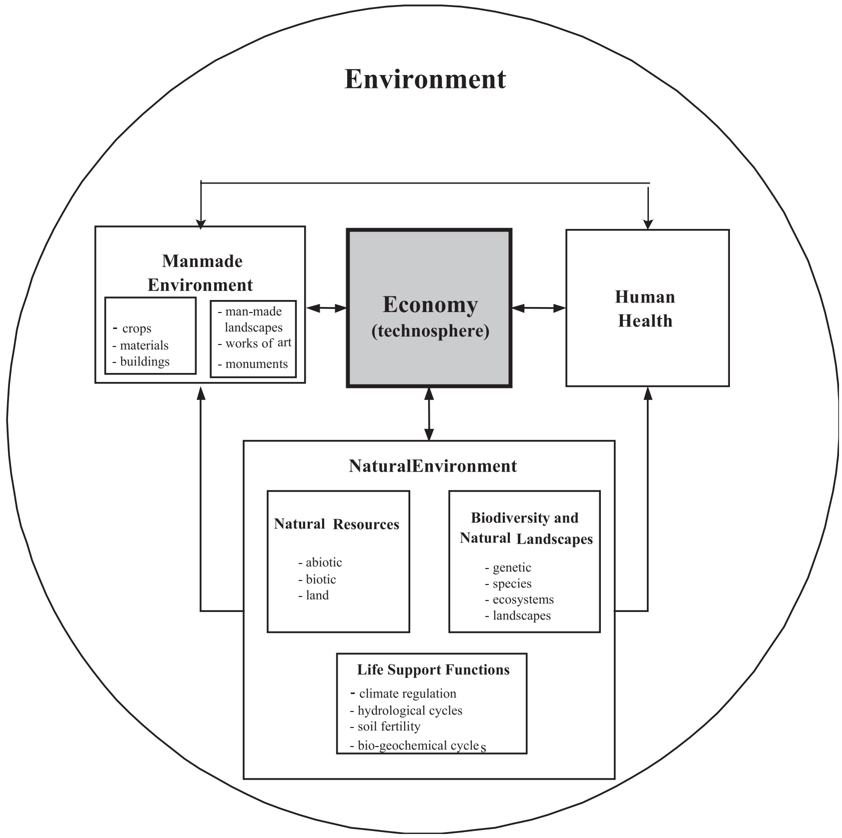

Fig. 3. Classification of AoPs according to societal values (Udo de Haes and Lindeijer, 2002).

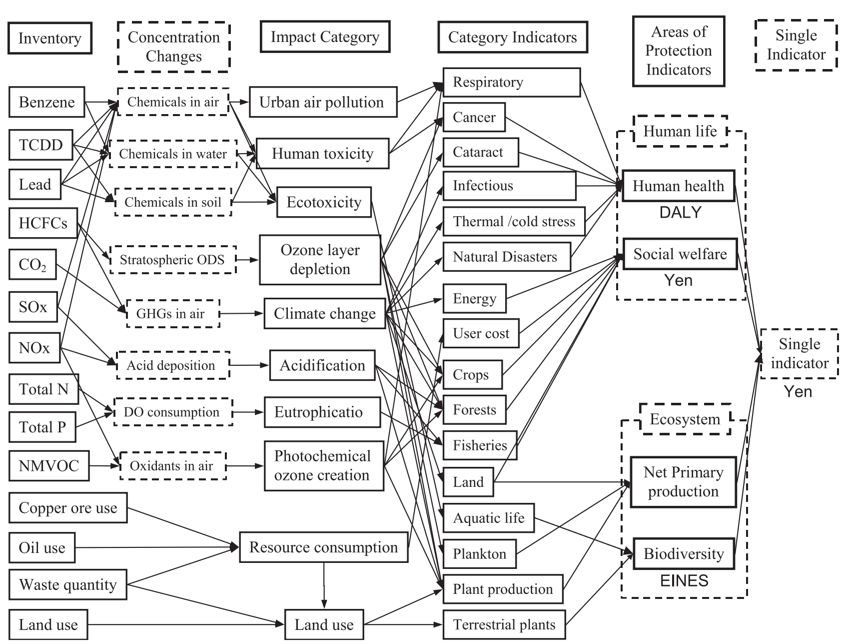

Fig. 4. Illustrative inventory data, category indicators, areas of protection, as well as an optional step to estimate a final indicator in terms of Yen. (Based on the Japanese national method "LIME", Itsubo and Inaba, 2003.)

The emissions inventory data are in terms of the mass released into the environment—such as 1 kg—per functional unit. The characterisation factors from Eq. (1) therefore linearly express the contribution to an impact category of a unit mass (1 kg) of an emission to the environment.

As an example, the relative contributions of different gases to climate change are commonly compared in terms of carbon dioxide equivalents using Global Warming Potentials (GWPs). A GWP_{500} of 100 implies that 1 kg of the substance has the same cumulative climate change effect as 100 kg of carbon dioxide during, in this case, a 500 year time period. Section 6 provides an overview of other available characterisation factor options for climate change, as well as for the other impact categories.

Eq. (2) illustrates some of the potential variables of non-generic characterisation factors in the context of impacts on human health and the natural environment (analogous equations are again available for resource consumption).

$$\text{Charaterisation Factor}(s,i,t) = \sum_j \frac{\text{Effect}(s,j,t)}{\text{Emission}(s,i)} = \sum_j \left(\frac{\text{Fate}(s,j,t)}{\text{Emission}(s,i)}\right) \cdot \left(\frac{\text{Exposure}(s,j,t)}{\text{Fate}(s,j,t)}\right) \cdot \left(\frac{\text{Effect}(s,j,t)}{\text{Exposure}(s,j,t)}\right) \quad (2)$$

Subscript s again denotes the chemical, i is the location of the emission, j is the related location of exposure of the receptor and t is the time period during which the potential contribution to the impact is taken into account. Other combinations of variables can exist, depending on the methodology adopted and the impact category of interest.

Models are available that can take into account all the different variables in Eq. (2) for different impact categories. However, LCA studies and often the underlying characterisation models have traditionally not considered, or have even ignored, many of these variables. As outlined in the next sections, differences between approaches and issues of debate therefore include “to what extent?”, “under which circumstances?” and “pragmatically how?” to best account for the location and time of the emissions, waste generated and resources depleted in an LCA, as well as the location and time period over which the contributions to different impacts should be taken into consideration (e.g. Potting et al., 1998a,b; Huijbregts, 1998a,b, 1999; Hofstetter, 1998; van den Berg et al., 1999; Hertwich, 1999; Nigge, 2000; Potting, 2000, in press; Hellweg, 2001; Hellweg et al., 2003; Potting and Hauschild, in press).

4. LCIA in the context of other applications

Many variations of human health and environmental risk/impact assessment methodologies have been developed to address different questions. As the questions can differ from those of LCA, so can the approaches and the answers (Potting and Klöpffer, 2001; Cowell et al., 2002; Olsen et al., 2001; Wegener Sleeswijk, 2001; Potting, 2000; Owens, 1997). It is therefore beneficial to briefly consider the relationship of other common risk/impact assessment methodologies with those of LCIA.

LCIA could be considered in the context of integrated assessment modelling (IAM). While there is no clear definition, nor distinction, from many other categories of policy support tools (Hettelingh et al., 2004), an integrated assessment model essentially combines knowledge from different disciplinary fields to help simultaneously analyse environmental problems and solutions. The modelling is typically in a policy context and socioeconomic indicators provide a comparable basis that facilitates evaluation of various scenarios. In current practice, these integrated assessment models typically focus on a limited number of impact categories such as acidification.

RAINS is one example of an integrated assessment model that was originally developed for assessing acidifying effects in a European context (Hettelingh et al., 2004; Alcamo et al., 1990). Building on networks of experts, in close collaboration with policy makers, RAINS significantly contributed to the UNECE Convention on Long-Range Transboundary Air Pollution. Potting et al. (1998a,b), and later Huijbregts et al. (2000c), directly used the RAINS model in an LCA context to establish region-dependent characterisation factors for acidification (see Section 6 for further examples). These factors are available for emissions from 44 regions (mainly countries) and provide estimates of the marginal increase in area that is exposed above critical threshold levels per unit mass of acidifying substance emitted.

Methodologies and models developed for assessments in other contexts can also differ from those of LCA. For example, methodologies for screening chemicals in a regulatory context provided a starting point for the development of some toxicological impact indicators for use in LCA. The results from the LCA do not provide measures of risk that can be directly compared to the traditional policy-based thresholds, or standards, of the underlying methodologies. The methods instead used the toxicological thresholds and standards to help compare chemical emissions in terms of “policy-based hazard equivalents” (Guinée et al., 1996; Hertwich, 1999; Huijbregts, 1999; Huijbregts et al., 2000a; Hertwich et al., 2001; Bare et al., 2003). The characterisation factors are interpreted as “hazard equivalents”—e.g. “kg equivalents of benzene” for cancer effects are the ratio of the hazard of a unit emission (kg/hour) of one chemical relative to that for a reference chemical, in this case benzene. As outlined in Section 6, alternatives are now emerging that are interpretable in terms of marginal cumulative risks (risks of effects that are integrated over time and space) (Crettaz et al., 2002; Pennington et al., 2002, submitted for publication, in press).

The exact reasons for differences between indicators in LCA and in regulatory contexts vary. The differences depend somewhat on the LCIA methods selected, as well as on the non-LCA approaches considered. The reasons can include:

- Wastes and emissions in a life cycle inventory for a particular site often will not reflect the full extent of

releases to the environment from that site. All the wastes/emissions from a site usually do not contribute to the provision of a product, as defined by the functional unit in an LCA study, nor are they necessarily associated with only the product of interest in the study. In LCA, depending on the approach adopted, the marginal contributions to risks and impacts can be estimated that are attributable to a product. This marginal approach primarily differs from regulatory assessment practices for toxicological risks. The full extent of emissions from a site, background levels and influences of mixtures are crucial in calculating absolute toxicological risks, hence, for assessing the likelihood of exceeding policy-based standards. But, checking for compliance and the consideration of such political standards are not relevant in an LCA context.
- LCA is a comparative assessment methodology. Inconsistencies in the assessment can introduce unintentional bias. Direct adoption of regulatory methodology and data is not always appropriate. Regulatory methods and data, again particularly in toxicological risk assessments, are not always developed for use in a comparative context. Best-estimates are desirable in LCA, with the often-overlooked need to account for uncertainties when making distinctions amongst the results.
- The underlying nature of the risks and impacts estimated in an LCA can differ from those of interest in other applications. As an example, some recent LCIA approaches for toxicological impacts attempt to estimate cumulative risks on a population basis (risks to the entire population that are integrated over time and space). This is not the same risk basis as in common regulatory toxicological impact approaches, where specific exposure concentrations for individuals are compared to policy-based toxicity thresholds or standards.

Unlike many other approaches, LCA often also provides simultaneous indicators related to many different impact categories (climate change, stratospheric ozone depletion, depletion of resources, toxicological effects, noise, etc.). The desirability and extent to which these indicators for the different impact categories can be integrated on a natural science basis varies, even within each AoP such as human health. How LCA frameworks facilitate integrated decision support therefore also varies, as outlined in the next sections.

5. Key modelling issues

Modelling in LCIA depends on the impact category, the basis of the chosen category indicator, the level of acceptable uncertainty or accuracy, expert judgement, different viewpoints and on the optional socioeconomic techniques that are sometimes adopted to help compare across indicators for different impact categories and AoPs. This section briefly outlines many of the associated modelling issues and their implications. Section 7 discusses further the issue of comparing across impact categories, or AoPs, using social science methods.

5.1. Marginal versus average impacts

Distinctions exist between three broad classes of comparative indicators in current LCIA practice:

- First generation indicators that contributed to weighting inventory data using policy-based measures or intrinsic properties. Indicator results are commonly interpretable in terms of "policy-based hazard equivalents" or acidification potentials based on a number of H+ ions, for example.
- *Marginal approaches* provide estimates of marginal changes, i.e. small changes, to the existing risks and (potential) impacts that would be attributable to a change in, or the provision of, different goods and services (Udo de Haes et al., 1999).
- *Average approaches* yield estimates of the contributions of a product to the overall status quo of risks and (potential) impacts (Udo de Haes et al., 1999).

The practitioner's choice between policy-based, marginal or average indicators depends somewhat on the goal and scope of their LCA study (as well as what approaches are available, as outlined in Section 6). The significance of the differences between average, marginal and policy-based factors is, however, the topic of ongoing research and discussion.

Fig. 5 illustrates the differences of average versus marginal approaches in the context of (eco)toxicological impact characterisation factors. Based on a comparison of underlying theories and assumptions, the marginal versus average indicator differences were considered to be more of an academic interest for toxicological impacts than of practical relevance (Crettaz et al., 2002; Pennington et al., 2002, 2004). The results using the marginal or average approaches can differ significantly, however, from the policy-based indicators for toxicological effects, while the results can be more strongly related to those of policy applications for other impact categories (as outlined earlier in Section 4).

5.2. Depth and breadth of mechanism modelling

Another key issue in LCIA is the extent to which environmental mechanisms (cause–effect chains) are modelled (Finnveden et al., 1992; Potting and Hauschild, 1997; Bare et al., 2002b; Udo de Haes et al., 2002). This is illustrated with the following example. Emissions of CO_2 lead to increased concentrations of CO_2 in the atmosphere, which in turn results in increased absorption of infrared radiation. Changes in absorption of infrared radiation associated with other chemicals provide a common (middle or mid-) point in the environmental mechanism. Comparisons

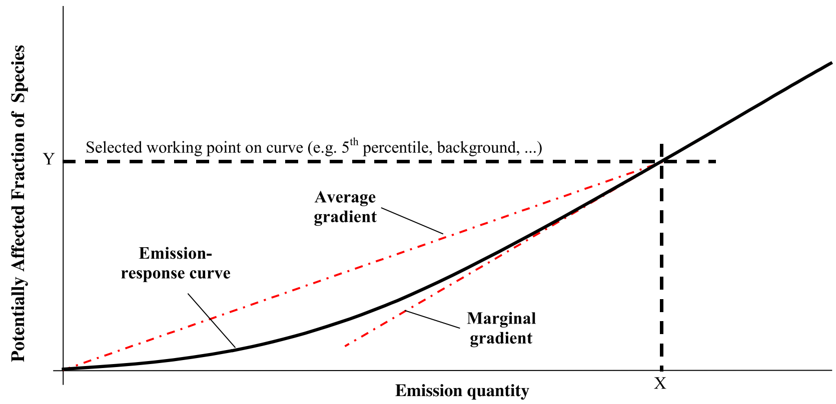

Fig. 5. Example of marginal and average characterisation factors (Δeffect/Δemission gradients) for ecotoxicological impacts (based on Pennington et al., 2004). The linear gradient, the gradient between a working point and the origin of the curve, is the average gradient. The tangential gradient at a specified working point on the curve is the marginal gradient.

are feasible at this common point. The climate change category indicator is then presented in terms of changes in infrared radiation absorption capacity using characterisation factors such as the GWPs that were outlined in Section 1.

Changes in infrared radiation are considered to cause changes in temperature, as well as other changes in the global climate. These changes have the potential to result in impacts on humans and on ecosystems (through flooding, increased cases of diseases such as malaria, changes in thermal stress, etc.) (McCarthy et al., 2001). It can also be desirable to explicitly model up to such effects. The result of this modelling is a set of indicators referred to as endpoint, damage or impact indicators in LCA circles. Such modelling is better established for some impact categories, while extensive debates and opinions revolve around others.

The advantages and disadvantages of choosing indicators early (sometimes termed midpoint indicators), or late (damage, endpoint or potential impact indicators) have been extensively discussed (Bare et al., 2002b). It was argued by some that choosing indicators late in the mechanism (closer to an endpoint) facilitates more structured and explicit weighting across impact categories based on natural science (Bare et al., 2002b; Udo de Haes et al., 2002). For example, it is then possible to directly compare the relative importance of toxicological impacts and climate change on the AoP of human health using a single indicator such as the disability adjusted life years (DALYs). One DALY represents 1 year of life lost, or an equivalent in the case of morbidity effects (Murray and Lopez, 1996; Hofstetter, 1998; Hofstetter and Hammitt, 2002).

Some stakeholders, however, still preferred using indicators earlier in an environmental mechanism for some impact categories, particularly for climate change, because of the additional uncertainties and forecasting considered necessary when modelling indicators closer to an endpoint. Socio-economic techniques then facilitate comparisons across the impact category indicators, and even across indicators for different AoPs such as human health and natural resources—if desirable (Bare et al., 2002b; Finnveden et al., 2002; Udo de Haes et al., 2002). Section 7 elaborates further on comparing across impact categories and AoPs using social science and economic techniques.

Based on a series of workshops, Bare et al. (1999, 2002a,b) concluded that, where relevant, it is preferable to provide indicators at both the midpoints and the endpoints using consistent modelling approaches. It was also noted that natural science-based insights for categories such as climate change provide useful information, irrespective of whether social science and economic techniques are used or not.

5.3. Site/temporal variability

A life cycle inventory can relate to a very large number of sites, locations and time periods. These attributes are largely, or often completely, unknown (as outlined in Part 1, Rebitzer et al., 2004). The degree to which the high temporal and spatial variability in some inventories eliminates the need for more specific factors is also generally unknown.

Current LCIA practice usually relies on site/temporal-generic characterisation factors. That is not to say that models lacking in temporal/spatial resolution are the best option for calculating such generic factors. Variables that could be taken into consideration include:

- the location and the time of generating waste, emissions and resource consumption (region or country of the emission, urban vs. rural location, artic versus tropical zone, into a lake versus a river, etc.)
- the mode of entry of an emission into the environment (from a tall stack versus low-level dispersion, to air, to surface water, to sea water, etc.)
- and/or the sensitivity of the different receiving/affected environments in terms of time and space.

Hauschild and Potting (in press), Potting (2000), Nigge (2000), Hertwich (1999), Potting et al. (1998a,b), Potting and Blok (1994) and others have provided preliminary

insights into the potential significance of some of these variables—primarily looking at variability in terms of location and for chemical emissions.

Site/temporal-*dependent* characterisation factors are proposed as perhaps providing a practical solution to help address the space/time issues in LCA. These factors are between the two extremes of site/temporal-*specific* and generic factors in terms of complexity (Nigge, 2000, Potting, 2000). While in many cases a single model can provide all three types of factors, identifying significant attributes and specifying the situations where such attributes need to be taken into account is of high practical importance when collecting, or refining, life cycle inventory data (Rebitzer et al., 2004).

5.4. Uncertainty

Uncertainties can increase as modelling is extended along the mechanism, for example from predicting increased absorption of infrared radiation to forecasting impacts on human health attributable to climate change, or as greater spatial and temporal resolution are taken into account.

Factors from simplistic modelling approaches may have lower *parameter* uncertainty than those from more complex models. Such simplistic models can be based on a small number of input parameters that are more readily available and may reflect only a small part of an environmental mechanism. But, when viewed in isolation, such lower parameter uncertainty can convey a false sense of confidence to a decision-maker.

For additional complexity to be justified, characterisation factors from complex models should be more accurate than those of the simplistic approaches and such accuracy must be necessary for attaining a reliable decision. A model of higher complexity, usually one that explicitly represents more of an environmental mechanism or one that allows for higher spatial/temporal resolution, can involve more explicit assumptions and may have higher input data requirements. The parameter uncertainty can then be higher—although this depends in practice on which parameters dominate the results and if these are common to both types of model. However, the results of the complex models could prove to have a lower overall uncertainty. The combined model and parameter uncertainty would be lower. The model uncertainty is the more intrinsic uncertainty associated with the underlying algorithms of the model, which is often not quantified using techniques such as Monte Carlo analysis.

Unfortunately, the trade-offs between model complexity, parameter uncertainty, and overall accuracy (model + parameter uncertainty) often remain unclear and un-quantified. A combination of reliance on the wishes of stakeholders, expert judgement, inter-model comparison insights, sensitivity analyses and the few model evaluations that are available all remain essential to judging the extent to which additional complexity and higher data demands are likely to be beneficial, or not.

6. Overview of existing models for common impact categories

As foreseen from the previous sections, a large number of indicators and supporting methodologies are feasible for estimating characterisation factors for the different impact categories and areas of protection in LCA. Entire methodologies were compiled in Wenzel et al. (1997), Hauschild and Wenzel (1998), BUWAL (1998), Goedkoop and Spriensma (1999), Steen (1999a,b), Guinée (2002), Itsubo (2000), Itsubo and Inaba (2003), Bare et al. (2003), Hauschild and Potting (in press) and Jolliet et al. (2003)—in addition to the standalone/sub-groups of methods and models that were developed elsewhere for specific impact categories.

Table 1 provides examples of the available methods for common impact categories in LCAs. A complimentary discussion, with additional details of the underlying principles, is available in Udo de Haes et al. (2002).

7. Normalisation, grouping and weighting

A few data in a life cycle inventory can often dominate within an impact category. The results are then readily interpretable. However, a practitioner may also want to compare across impact categories, or even areas of protection, to prioritise or to resolve trade-offs between product alternatives (e.g. lower climate change indicators for one option, but higher toxicological indicator results for another). This can be achieved, to some extent, using natural science approaches—particularly within areas of protection such as human health. But, limitations remain.

Comparing across impact category indicators is an optional step in some applications of LCA (ISO 14042, 2000). In common LCA practice, this optional step draws not only on natural sciences, but often relies heavily on social science and, in some cases, on economics. The following subsections introduce some of the techniques for normalisation, grouping and weighting.

7.1. Normalisation

The aim of normalisation is typically two-fold (Finnveden et al., 2002):

- to place LCIA indicator results into a broader context and
- to adjust the results to have common dimensions.

The sum of each category indicator result is divided by a reference value,

$$N_k = S_k / R_k \tag{3}$$

Table 1
Illustrative background summary of LCIA category indicators, characterisation factors and models (not exhaustive). The level of scientific review that these different methods have undergone varies significantly and no indication of preference is suggested. This list is not intended to be exhaustive in terms of methods nor impact categories

Category indicator basis	References
Climate change	
Increase of radiative forcing (GWPs)	IPCC (Nakicenovic and Swart, 2000)
Increase of human health effects (malaria, schistosomiasis, dengue fever, cardiovascular and respiratory disorders, displacement of people due to sea level rise)	Goedkoop and Spriensma (1999)
Increase of human health effects (heat/cold stress, malaria, dengue fever, disaster, nutrient deficiency)	Itaoka et al. (2002)
Increase in crop and primary production damage	Uchida et al. (2002)
Increase of several damages—human health and ecosystem effects (heat stress, malaria, starvation, flooding accidents, wood growth, contribution to species extinction)	Steen (1999a,b)

Heat is trapped in the atmosphere by the infrared adsorption of reflected sunlight in a spectral window between ~ 10 and 15 μm. Changes in this adsorption capacity can result in changes in the earth's climate. Anthropogenic emissions contributing significantly to this capacity include carbon dioxide, methane and nitrous oxide—all examples of "greenhouse gases".

The relative contributions of different chemical emissions to climate change are commonly calculated in LCIA using GWPs. GWPs are referenced to 1 mass unit of carbon dioxide. For example, a GWP_i, 500 of 100 implies that 1 kg of chemical (i) has the same predicted climate change effect, over the example time horizon of 500 years, as 100 kg of carbon dioxide. The Intergovernmental Panel for Climate Change (IPCC) proposed a time horizon of 100 years, whilst also providing GWPs for different time horizons.

Enhanced radiative forcing and climate change can both be considered primary effects. The potential consequences of climate change could be dramatic (McCarthy et al., 2001). Research is ongoing to predict these consequences. Goedkoop and Spriensma (1999), Steen (1999a,b), Itaoka et al. (2002), and Uchida et al. (2002), for example, used associated models and literature to calculate characterisation factors in terms of these potential consequences—mainly in the context of human health. These factors help to facilitate direct cross-comparison with other impact categories at an area of protection level, as outlined in Section 5. However, given the additional modelling and forecasting, these predictions are considered more uncertain than the GWPs and not all potential consequences are necessarily taken into account.

Category indicator basis	References
Stratospheric ozone depletion	
Increase of ozone breakdown (ozone depletion potentials, ODPs)	WMO (1991, 1999)
Increase in human skin cancer	Steen (1999a,b)
Increase of several damages—human health—skin cancer and cataracts	Goedkoop and Spriensma (1999), Hayashi et al. (2000a, 2002)

Stratospheric ozone depletion refers to the loss of the ultraviolet (UV) absorption capacity through the destruction of ozone in the stratosphere. UV absorption hinders radiation below 300 nm from reaching the earth's surface.

ODPs express the ozone depleting capacity of chlorofluorocarbons (CFCs), hydrochlorofluorocarbons (HCFCs), and halons relative to the reference substance CFC-11, mainly related to their ability to release chlorine and bromine radicals in the stratosphere (World Meteorological Organization, 1991).

Some of the consequences of increased short-wavelength UV radiation at the earth's surface are known, especially skin cancer formation. Steen (1999a,b), Goedkoop and Spriensma (1999) and Hayashi et al. (2000a, 2002) used the associated literature and models to calculate characterisation factors in terms of such consequences. These factors help facilitate cross comparison with the consequences attributable to other impact categories (see Section 5). However, as additional modelling is required and they do not cover all effects from ozone depletion (particularly effects on ecosystems), these factors are considered more uncertain than the ODPs.

It should be noted that many of the most problematic ozone depleting chemicals are now banned, as a consequence of the Protocol of Montreal in 1987, subsequent adjustments and amendments (WMO, 1999). The importance of this impact category in LCA may therefore also be diminishing in some application contexts.

Category indicator basis	Spatial attributes		References
	Site-generic only	Site-dependent	
Acidification			
Release of hydrogen ions (acidification potentials)	Yes	No	Wenzel et al. (1997), Heijungs et al. (1992), Lindfors et al. (1995)
Increase of acid deposition on land surface	Yes	Yes	Bare et al. (2003), Norris (2003)
Increase of hazard index for ecosystems above critical levels	Yes	Yes	Huijbregts et al. (2000c), Guinée (2002)
Increase of ecosystem area exposed above critical threshold levels	Yes	Yes	Potting et al. (1998a,b), Krewitt et al. (2001)
Increase of potential disappeared fraction of plant species	Yes	No	Goedkoop and Spriensma (1999)
Reduction in terrestrial plant primary production	Yes	No	Hayashi et al. (2000b)
Increase of several impacts (to fish production capacity, base cation capacity of soils, contribution to species extinction)	Yes	No	Steen (1999b)

Acidification refers to an increase in acidity, the hydrogen ion concentration, in water and soil systems. LCA has often adopted the number of hydrogen ions that are theoretically formed per unit mass of a chemical as a basis for characterisation. The characterisation factors express an emission in terms of sulphur dioxide equivalents (Wenzel et al., 1997; Heijungs et al., 1992; Lindfors et al., 1995). This simplistic approach was criticised as being a potentially poor basis for environmental impacts of acidifying pollutants, given the disregard of differences related to source location, atmospheric transport capabilities and the relative sensitivity of receiving environments (Potting and Blok, 1994).

Potting et al. (1998a,b), and in a different way Huijbregts et al. (2000c) as adopted in Guinée (2002), used the RAINS-model (Alcamo, 1990) to calculate region-dependent characterisation factors for acidification for Europe. Krewitt et al. (2001) established similar factors as Potting et al. (1998a,b) using the EcoSense integrated assessment model. These factors relate an emission in a given region (typically country level) to changes in terms of the receiving ecosystem area that is exposed above critical levels (levels above which exposure can exceed the carrying capacity of the environmental media and then influence the ecosystems). All these methods suggest that disregarding spatial variability by using generic factors could lead to significant increases in uncertainty in the results of some LCA studies (potentially a factor of thousand in some cases).

Norris (2003), as adopted in Bare et al. (2003), proposed similar spatially resolved factors for the US—although only accounting for variations in deposition and not for critical effect thresholds.

Steen (1999a,b) provided factors based on data in the literature without spatial resolution but that take into account for the effects of acidification. The effects are quantified in terms of reduced fish production capacity, reduced base-ion capacity of soils and contributions to the extinction of species based on overall threats to endangered species. Goedkoop and Spriensma (1999) established similar factors for calculating damage to flora, using a combined approach for the effects of acidification and terrestrial eutrophication. The Goedkoop and Spriensma (1999) damage factors are derived from the highly spatially resolved Natuurplanner model for the Netherlands (Kros et al., 1995). However, the Netherlands is too small to serve as a basis for calculating spatially resolved characterisation factors for acidification, while the susceptibility and background loading of Dutch ecosystems are probably not representative for other regions. Hayashi et al. (2000b) provided similar acidification factors for flora for Japan.

Category indicator basis	Spatial attributes		References
	Site-generic only	Site-dependent	
Aquatic eutrophication			
Increased growth of aquatic biomass	Yes	No	Heijungs et al. (1992), Samuelsson (1993), Wenzel et al. (1997)
Increased growth of aquatic biomass	Yes	Yes	Huijbregts and Seppälä (2000), Huijbregts and Seppälä (2001), Seppälä et al. (in press)
Increased growth of aquatic biomass in inland and marine waters	Yes	Yes	Potting et al. (2003a), Norris (2003)
Several effects (increased fish production capacity, contribution to species extinction)	Yes	No	Steen (1999b)
Several effects (fisheries and biodiversity)	Yes	Yes	Hirosaki et al. (2002)

Aquatic eutrophication is the result of nutrient enrichment in aquatic environments. Under natural conditions, the supply of nutrients to water is in balance with the growth of biomass. Anthropogenic nutrient inputs can disturb this balance, leading to increases in algal growth that make the water turbid and decrease the level of oxygen content. This then leads, for example, to increases in fish mortality and ultimately the disappearance of bottom fauna (Kristensen and Hansen, 1994).

LCA commonly takes nutrient emissions as a basis for characterisation factors of aquatic eutrophication. Characterisation factors are often based on the relative ratio of the phosphorus (P) and nitrogen (N) nutrients in the composition of phytoplankton (C/N/P = 106:16:1, the Redfield-ratio) (Stumm and Morgan, 1981; Redfield et al., 1963).

Oxygen demand associated with the degradation of organic matter is often related to eutrophication in LCA. However, increases in COD or BOD do not necessarily result in biomass growth, as this is primarily associated with changes in nitrogen and phosphorus levels. While being related in terms of consequences, indicators for increases in COD or BOD thus reflect different cause–effect relationships to indicators for nutrient enrichment (Finnveden et al., 1992; Samuelsson, 1993; Lindfors et al., 1995; Finnveden and Potting, 1999; Potting et al., 2003a).

Many characterisation factors for eutrophication do not account for spatial variability associated with source location, environmental transport and ecosystem sensitivity. Several sets of region-specific factors have therefore been proposed in the last few years (Norris, 2003, as adopted in Bare et al., 2003; Seppälä et al., in press; Potting et al., 2003a; Hirosaki et al., 2002; Huijbregts and Seppälä, 2000, 2001). Distinctions are made between P-limiting, usually freshwater, and N-limiting, usually seawater, environments. Nevertheless, these factors only account for the transport of nutrients through air and from soils to water. They do not address spatial variations in terms of the effects.

Based on models and data in the literature, Steen (1999b) and Hirosaki et al. (2002), for example, proposed factors that account for the impacts associated with aquatic nutrient enrichment, as well as COD/BOD loading. These factors are expressed in terms of changes in fish production capacity and the potential contribution to the extinction of (endangered) species due to aquatic eutrophication.

Category indicator basis	Spatial attributes		References
	Site-generic only	Site-dependent	
Terrestrial eutrophication			
Increase of hazard index for ecosystems above critical levels	Yes	Yes	Huijbregts et al. (2000c), Guinée (2002)
Increase of ecosystem area exposed above critical levels	Yes	Yes	Potting (in press), Potting et al. (2003b), Krewitt et al. (2001)
Increase of potential disappeared fraction of vegetation species (dealing with terrestrial eutrophication and acidification together)	Yes	No	Goedkoop and Spriensma (1999)

(continued on next page)

Several effects (increase of wood growth capacity, contribution to species extinction)	Yes	No	Steen (1999b)

Terrestrial eutrophication is associated with the nutrient enrichment of soils. The growth of plants is commonly controlled by the limited availability of nitrogen (phosphorus seldom limits plant growth in terrestrial ecosystems). Exposure of nitrogen-limited ecosystems to increased nitrogen loads often increases the competitive advantage of nitrogen-adapted species at the expense of other species. This alters the tolerance of ecosystems towards disease, drought, frost, and herbivores (Hornung et al., 1994; Grouzet et al., 1999).

Considering spatial distinction of both the emission source as well as receptor region properties, Potting (in press), Potting et al. (2003b), Krewitt et al. (2001), Huijbregts et al. (2000c), and Guinée (2002) have established region-specific, as well as generic, characterisation factors for terrestrial eutrophication. Currently, based on the scope of the models and not necessarily reflecting an optimal approach, these are available for 44 European regions. The factors reflect the change in the area affected above critical levels.

Although not accounting for spatial differentiation, Goedkoop and Spriensma (1999) and Steen (1999b) provided characterisation factors that facilitate cross comparison with other impact category indicators by accounting for the consequences of changes in exposure. Goedkoop and Spriensma's (1999) factors address acidification and terrestrial eutrophication together. Steen (1999b) factors were developed from empirically estimated contributions to wood growth capacity and species extinction that were attributed to terrestrial eutrophication (favouring nitrogen-adapted species).

Category indicator basis	References
Human toxicological effects	
Scores based on physical–chemical fate properties and policy-based toxicological thresholds	Hauschild and Wenzel (1998)
Policy-based hazard equivalents using multimedia fate and multi-pathway exposure modelling with policy-based toxicological thresholds	Guinée et al. (1996), Hertwich (1999), Hertwich et al. (2001), Huijbregts (1999), Huijbregts et al. (2000a), Bare et al. (2003)
Cancer and respiratory factors, accounting for impacts in terms of DALY	Hofstetter (1998), Steen (1999b), Goedkoop and Spriensma (1999), Nagata et al. (2002), Sakao et al. (2002), Crettaz et al. (2002), Pennington et al. (2002, submitted for publication), Bare et al. (2003)
Marginal change in toxic effects associated with cumulative individual human inhalation exposure per kg emission. Emission–receptor relationships based on atmospheric transport models	Hauschild and Potting (in press), Potting (2003), Krewitt et al. (2001)
Marginal change in cumulative population-based risk and potential impacts (DALYs) per kilogram emission using multimedia/multi-pathway models with low dose extrapolation from toxicological benchmarks and potential for spatial source-to-receptor resolution	Pennington et al. (2002, submitted for publication), Crettaz et al. (2002)

A toxicological effect is an adverse change in the structure, or function, of a species, in this case humans, as a result of exposure to a chemical.

Hauschild and Wenzel (1998) presented score-based factors that rank emissions in terms of selected fate and policy-based toxicity data. Guinée et al. (1996), Hertwich (1999), Hertwich et al. (2001), Huijbregts (1999), Huijbregts et al. (2000a) and Bare et al. (2003) calculated generic factors using mechanistic models. The factors can be interpreted in terms of "policy-based hazard equivalents". For example, indicators could be presented in terms of "kg equivalents of benzene" for cancer effects, reflecting the ratio of the hazard of a unit emission (kg/h) of one chemical relative to that of benzene. Calculations are performed using multimedia chemical fate models, human exposure correlations for organic chemicals and toxicological methodologies/data designed for chemical risk screening in a regulatory context. Pennington et al. (2003b) and Dreyer et al. (2003) provided a comparison between such score-based and model-based approaches for LCIA.

Pennington et al. (submitted for publication) proposed a methodology to estimate characterisation factors in terms of the marginal change in cumulative cancer and non-cancer risks (integrated over time and space, as well as over the entire exposed population). The low-dose extrapolations are based on a toxicological benchmark approach. Benchmarks are an exposure measure associated with a consistent change in response, such as the 10% effect level (Crettaz et al., 2002; Pennington et al. 2002).

To additionally account for differences in the severity of potential human health effects, characterisation factors can be expressed in terms of metrics such as DALYs (Murray and Lopez, 1996, Hofstetter, 1998; Goedkoop and Spriensma, 1999; Nagata et al., 2002; Crettaz et al., 2002; Hofstetter and Hammitt, 2002; Pennington et al. 2002, submitted for publication). These can be directly cross-compared with DALY-based indicators from other impact categories, such as climate change, to provide overall indicators for the human health area of protection. Metrics such as DALY account for mortality, and using social sciences to help provide equivalents in years of life lost, for morbidity (non fatal) consequences. Further development for non-cancer effect consequences are necessary (Pennington et al., 2002, submitted for publication).

In Hofstetter (1998) and Bare et al. (2003), some DALY-based factors are directly calculated from epidemiological data, including for secondary particulate matter (nitrates and sulphates). This reduces reliance on modelling results.

Complimentary, or as an alternative, to using quantitative metrics such as DALY, Owens (2002) illustrated the feasibility of a category-based approach for classifying hazard/risk-based factors according to different toxicological endpoints. A number of researchers have also started to address the influences of many other underlying modelling options on characterisation factors for toxicological effects. These include differentiation in terms of the mode of entry and location of a chemical release into the environment (Potting, 2000; Nigge, 2000; Krewitt et al., 2001; Potting, 2003; Pennington et al., submitted for publication), the duration over which the effects are likely to occur (discounting) in terms of time horizons like 50 or 500 years (Huijbregts et al., 2001, Hellweg, 2001, Hellweg et al., 2003), and uncertainty (Hofstetter, 1998; Huijbregts, 1998a,b, 1999; Hertwich et al., 1999; Thissen, 1999; Huijbregts et al., 2000b).

Category indicator basis	References
Ecotoxicological effects	
Scoring based on physical–chemical fate properties and ecotoxicological policy thresholds (predicted no effect concentrations, PNECs)	Hauschild and Potting (in press), Hauschild and Wenzel (1998)

Table 1 (*continued*)

Policy-based hazard equivalents for species in different environmental media (surface and oceanic waters, soils and sediments)	Guinée et al. (1996), Huijbregts (1999), Huijbregts et al. (2000a), Bare et al., 2003
Marginal risk-based factors expressed in terms of potentially affected fraction of species (PAFs) and the area or volume of surface water affected	Goedkoop and Spriensma (1999), Huijbregts et al. (2002), Payet and Jolliet (2003), Pennington et al. (in press)
Contribution to extinction of species (change in biodiversity)	Steen (1999b), Sakao et al. (2002)

An ecotoxicological effect is an adverse change in the structure, or function, of a species as a result of exposure to a chemical. Characterisation factors are best developed for aquatic species and in relation to chronic exposures, noting that more available acute toxicological data are often extrapolated to estimate such factors.

Hauschild and Wenzel (1998) and Hauschild and Potting (in press) presented score-based approaches that rank emissions in terms of fate and toxicity parameters that were judged to be important. Guinée et al. (1996), Huijbregts (1999) and Huijbregts et al. (2000a), for example, proposed factors based on mechanistic modelling and established policy-based risk measures/procedures (PNECs). The underlying principles and differences are analogous to those of the approaches outlined above for human toxicological effects.

Goedkoop and Spriensma (1999) provided factors in the context of PAFs (as described by Klepper and van de Meent, 1997). These factors reflect the marginal change in PAF per unit emission of a chemical given background effect level assumptions of 10–50% PAF. These factors draw on the concepts of species sensitivity distributions and concentration addition in mixtures (Hamers et al., 1996; Posthuma et al., 2002; Escher and Hermens, 2002). Huijbregts et al. (2002) illustrated some potential extensions to these concepts to account for response addition in mixtures.

Pennington et al. (in press) provided a review of PAF-based proposals and the underlying principles, demonstrating that model uncertainties will be high and that defendable differences between various options are primarily associated with the underlying toxicity data. A marginal risk-based effect factor, such as $\Delta PAF_{ms} = 0.5 \times \Sigma\Delta C/HC50$, the change in the potentially affected fraction of species that experiences an *increase* in stress for a change in exposure was suggested as probably the best available practice. Payet and Jolliet (2003) presented the AMI methodology to estimate such PAF-based measures for use in LCA, including parameter (data) uncertainty indicators and with data requirements equivalent to those of current regulatory approaches.

As polluting a small lake versus polluting all the lakes in a region to the same level of risk, for example, is not considered equivalent, factors for toxicological effects in LCA are multiplied by surface area of water affected in Goedkoop and Spriensma (1999). Pennington et al. (submitted for publication) and Payet and Jolliet (2003) multiply by the volume of water affected for aquatic impacts.

Beyond representing species assemblages, how exposure above the different effect levels (e.g. EC_{50}, NOEC) in the PAF-based approaches relates to impacts on the structure and the function of ecosystems remains somewhat unclear (Posthuma et al., 2002; Udo de Haes et al., 2002; Pennington et al., in press). Steen (1999b) estimated the contribution to extinction of species from chemical emissions using empirical insights in the literature for overall threats to so-called red-listed species. Based on conservation biology to estimate the risk of species extinction, Sakao et al. (2002) estimated factors in terms of biodiversity change—biodiversity change being one potential option for an ecosystem structure-related indicator. Such "damage-orientated" indicators may facilitate direct comparison across impact categories, such as land use versus ecotoxicological effects (noting that land-use effects could also be represented in terms of PAFs).

No current LCIA method accounts for the spatial dependency of the toxicological effects, including for human health, although distinctions can be made in the context of chemical fate and exposure.

Category indicator basis	Spatial attributes		References
	Site-generic only	Site-dependent	
Photooxidant formation			
Increase of ground-level ozone formation (maximum incremental reactivity, MIR)	Yes	No	Finlayson-Pitts and Pitts (1993)
Increase of ground-level ozone formation (photochemical ozone creation potentials, POCPs)	Yes	Limited	Derwent et al. (1996, 1998), Andersson-Sköld et al. (1992)
Increase of ground-level ozone formation	Yes	Yes	Potting et al. (1998a,b), Krewitt et al. (2001), Bare et al. (2003), Norris (2003)
Increase of human respiratory disease	Yes	No	Goedkoop and Spriensma (1999)
Increase in human respiratory disease	Yes	Yes	Nagata et al. (2002)
Increase of human exposure above predefined policy thresholds	Yes	Yes	Potting and Hauschild (in press)
Increase of several damages—lethal and sub-lethal health effects from ozone exposure and crop growth reduction	Yes	No	Steen (1999b)

Photochemical oxidant creation, or photooxidant formation, refers to the mixture of ozone and intermediate reaction products, such as peroxyacetyl nitrate (PAN), that are formed in the lower atmosphere under the influence of solar radiation in the visible and near-UV spectral ranges. The photochemical oxidation process depends on the presence of nitrogen oxides, OH-reactive hydrocarbons, and carbon monoxide. While formed through secondary reactions and commonly being treated separately, modelling the impacts of ozone is analogous to that of other air pollutant toxicological effects described above.

Finlayson-Pitts and Pitts (1993) used the MIRs to calculate generic characterisation factors relative to a reference substance for the U.S. The European counterpart of the MIR is the POCP of Derwent and Jenkin (1991), Andersson-Sköld et al. (1992) and Derwent et al. (1996, 1998). POCPs relate chemical emissions in equivalents of ethylene in the context of their ozone creation ability and are directly adopted in some LCA applications.

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Table 1 (*continued*)

Potting et al. (1998a,b) and Potting and Hauschild (in press) used the RAINS model (Alcamo, 1984) to estimate marginal and region-dependent characterisation factors for Europe. These take into account current emission levels and natural background levels, atmospheric transport, so-called critical ozone levels, as well as receptor density. Krewitt et al. (2001) established similar spatially resolved factors for Europe using the EcoSense integrated assessment model. Nagata et al. (2002) proposed spatially resolved factors for Japan. Norris (2003), as adopted in Bare et al. (2003), combined modelling and the literature insights to provide spatially resolved factors for the U.S. including for emissions of nitrogen oxides.

The differences amongst the various generic factors, such as the POCPs of Derwent et al. (1996, 1998), are small compared to differences among the spatially resolved factors (Hauschild et al., 2003). This suggests that accounting for the location of emissions may be more significant than distinguishing between different organic compounds in terms of ozone creation in LCA studies.

In addition to modelling the relative ozone creation potential of chemicals, although using different methods, Steen (1999b), Goedkoop and Spriensma (1999), Nagata et al. (2002) and Potting and Hauschild (in press) provided factors that also account for potential consequences of ozone exposure. This facilitates cross-comparison with the effects associated with other impact categories, although additional modelling was required and therefore uncertainties may be higher than with the ozone exposure-based methods.

Category indicator basis	References
Biotic resources	
From the perspective of resources for humans:	
– extraction rate, regeneration rate and stock	Guinée et al. (1996), Guinée (2002)
– reproduction time and stock	Sas et al. (1997)
– production capacity of crop, wood, fish and meat	Steen (1999b)
From the ecosystem perspective, accounting for extraction rate and potential extinction risks	Steen (1999b), Müller-Wenk (2002a)

Biotic (or living) resources can be considered from two key perspectives: their availability for future generations (AoP natural resources), and their contributions to ecosystem functioning and biodiversity (in the AoP of natural environment). Biotic resources are renewable, but overexploitation can lead to damages such as population decrease and eventually species extinction. Site specific/dependent approaches have not yet been adopted for this impact category.

For impacts on the availability of resources from a human perspective, the annual extraction rate minus the annual replenishment rate divided by the square of the current stock is proposed as a characterisation basis in Guinée (2002) (analogous to the reserves and extraction rate methods for abiotic resources above). This approach is operational, but for only a few biotic species in Guinée et al. (1996). According to Müller-Wenk (Lindeijer et al., 2002a), a factor to take recovery time into account should also be added.

For impacts on the natural environment, the difference between annual (global) extraction levels and the level at which there is no longer a threat of extinction is proposed as an indicator basis, using IUCN (2001) data (Müller-Wenk, in Lindeijer et al., 2002a)—but not yet operational. Sas et al. (1997) expressed the risk of extinction in terms of reproduction time and the current amount of biomass—currently operational for some wood and fish species. Similarly, the reduction in biotic resource production capacity (crop, wood, fish and meat) is one of the endpoint effects in Steen (1999a,b). This applies both if the cause is the direct use of the resources, as well as if the reduction is caused by pollutants through several pathways. The potential extinction rate is estimated through the probability of extinction of red-listed species Steen (1999a).

Category indicator basis	References
Abiotic resources	
Future recycling (modelling in inventory)	Pederson (1991)
Exergy consumption (or entropy production)	Finnveden and Östlund (1997)
Reserves and extraction rates	Wenzel et al. (1997)
Ultimate reserves and extraction rates	Guinée (2002)
Ore concentration trends	Goedkoop and Spriensma (1999)
Future extraction energy	Müller-Wenk (1999)
Acquisition of near-sustainable alternatives (e.g. rapeseed oil, biogas, etc., for fossil fuels)	Steen (1999b)
Water availability and scarcity	Cowell (1998), Bauer and Zapp (in press)

Non-living, or abiotic, resources are addressed in terms of their availability for present and future generations. In terms of competition, the SETAC-Europe working group (Lindeijer et al., 2002a) considered this impact category to be covered through pricing. In terms of depletion, available impact methods are based on the amounts of the deposits (their abundance), on the extraction rates (Wenzel et al., 1997; Guinée, 2002), on modelling (interventions due to) future ore extractions (Pederson, 1991; Goedkoop and Spriensma, 1999; Müller-Wenk, 1999; Steen, 1999b), or on exergy consumption (Finnveden and Östlund, 1997).

Abiotic flow resources (e.g. solar radiation, wind, tides/currents, rain/river water flows and land use over time) cannot be depleted. Lindeijer et al. (2002a) considered competition, as well as inefficient consumption, to be socioeconomic issues. However, the consumption of flow resources may be addressed without additional impact assessment steps in LCA (e.g. Cowell, 1998 for solar radiation).

The availability of deposit stocks (stocks that are irreversibly depleted) to future generations is very uncertain. Modelling future extractions is similarly uncertain. Substitutability and accumulation in the economy are not currently taken into account (Lindeijer et al., 2002a). For non-ore abiotic resources, Steen (1999b) modelled the abiotic resource indicators in terms of the impacts for acquisition of a "near-sustainable alternative". For fossil oil, gas and coal, these alternatives are rapeseed oil, biogas, and charcoal, respectively.

For fund resources (e.g. water buffers and mineral soils that can be temporarily depleted) some site-generic proposals exist (e.g. Cowell, 1998). Freshwater can also be a flow, as well as a fund, resource (Bauer and Zapp, in press). The exceptional case of using fossil groundwater is considered as depletion of a stock.

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Table 1 (*continued*)

Category indicator basis	Spatial attributes		Separate transformation[a]	References
	Site-generic only	Site-dependent		
Land use impacts				
From a biodiversity perspective:				
–biological accumulation (standing biomass/GPP)	Yes	No	No	Knoepfel (1995)
–vascular plant species density and red list species	Yes	No	No	Felten and Glod (1995)
–vascular plant species density and red list species density	Yes	No	No	Köllner (2001)
–biome vulnerability, scarcity and original species quality	No	Yes	No	Weidema (2001)
–species density, biome vulnerability, scarcity and quality	No	Yes	Yes	Lindeijer et al. (2002b)
–increase of potentially disappeared fraction of vascular plant species	Yes	No	No	Goedkoop and Spriensma (1999)
–increase of extinction risk for vascular plants	Yes	Yes	Yes	Nakagawa et al. (2002)
–contribution to extinction of species	Yes	No	No	Steen (1999b)
From a life support perspective:				
–fNPP (free net primary biomass productivity)	No	Yes	Yes	Lindeijer et al. (1998)
–NPP (net primary biomass productivity)	No	Yes	Yes	Lindeijer et al. (2002b)
–NPP (net primary biomass productivity)	Yes	Yes	Yes	Nakagawa et al. (2002), Bauer and Zapp (in press)
–various: erosion resistance, filter/buffer/transformation—and ground water function, run-off regulation, biotic carrying capacity, emission protection, ecotope formation.	No	Yes	No	Baitz (2002)

Many methods for addressing land use in LCA have been proposed since 1992, but development towards characterisation factors started in 1995 (Lindeijer, 2000). Land use is currently divided into two types[a]: (1) land occupation (continuous use for the same process types, in units of area multiplied by time [m2.y]), and (2) land transformation (net change from one land use type to another, or changing quality, in units of area [m^2]). Competition for land is not considered useful as a separate impact indicator basis, according to Lindeijer et al. (2002a), although Guinée (2002) proposed direct addition of occupied land as an indicator.

Occupation and transformation relate to different impacts. Occupation leads to positive, or negative, impacts in terms of the stress on surrounding ecosystems, and can prevent restoration processes relative to some equilibrium or baseline. Transformation leads to net positive, or negative, impacts on the transformed area itself. The sign of the impact depends on the baseline, or on the land use just before transformation, respectively.

More regional and land use class specific approaches have been suggested, for instance in Sweden by Blümer and Kyläkorpi (1998), in Germany by Giegrich and Sturm (1996) and Schweinle (2001), in the US by Trusty and Paehlke (1997), in Japan by Nakagawa et al. (2002) and Bauer and Zapp (in press) on a global scale—but these are not applicable for all land uses. Different methodological choices are possible and the best solutions to deal with these have yet to be agreed (Lindeijer et al., submitted for publication).

Impacts of water extraction and other interventions that are related to land use may also be expressed in terms of land use impacts. For example, groundwater extraction impacts on ecosystems have been studied by van Ek et al. (2002), resulting in a method for the Netherlands and some proposals for site-dependent approaches for elsewhere. Similarly, Huber et al. (2002) proposed an approach for impacts of waterbed alterations on ecosystems.

[a]According to the definition in Lindeijer et al. (2002a). Ignoring transformation leads to excluding irreversible, or at least not reversed, impacts due to land use changes, such as degradation of tropical rainforests and the continuous expansion of urban areas.

Category indicator basis	References
Other physical interventions	
Odour	Guinée (2002)
Ionisation damage as years/kBq and as DALY	Frischknecht et al. (2000)
Ionisation damage as DALY	Goedkoop and Spriensma (1999)
Nuisance from odour and traffic noise	Steen (1999b)
Noise as DALY	Müller-Wenk (2002b)

A few methods exist addressing impacts from other physical interventions than those mentioned above. These include, but are not limited to:

For odour, Dutch malodourous air thresholds from 1989 (based on panels) are proposed as a basis for factors in Guinée (2002).

In the same reference, the damage factors for ionisation of Frischknecht et al. (2000) are preferred as a baseline method (in terms of years/kBq), with additional screening factors from Solberg-Johansen (1998). Goedkoop and Spriensma (1999) adopted ionisation factors in terms of DALYs (DALYs being described under human toxicological effects) from an earlier version of Frischknecht et al. (2000). Adopting DALYs facilitates cross comparison with other human health related impact category indicators.

Müller-Wenk (2002b) proposed factors in terms of DALYs for traffic noise, drawing on inputs from a medical expert panel.

where k denotes the impact category, N is the normalised indicator, S is the category indicator from the characterisation phase and R is the reference value.

The reference system is generally chosen using overall indicator results for a specific region, for example a country, and for a specific year, such as the annual national US contribution to climate change in terms of GWPs. Spatial scale, temporal scale, a defined system (e.g. a region or an economic sector) and a per capita basis are all examples of attributes that could be taken into account when choosing the reference value.

Normalisation results can provide input to grouping or weighting, as described in the next subsections, or can help directly judge the relative importance of different impact categories within an LCA study. However, it should be noted that direct application implies acceptance of the ratios of different impacts as they exist today, meaning that, for example, the total current effects of global warming and ecotoxicological effects in Europe would be considered to be of equivalent importance.

7.2. Grouping

Grouping is a qualitative, or semi-quantitative, process that involves sorting and/or ranking results across impact categories. Grouping may result in a broad ranking, or hierarchy, of impact categories with respect to their importance. Such a ranking can provide structure to help draw conclusions on the relative importance of different impact categories. For example, categories could be grouped in terms of high importance, moderate importance and low priority issues. Some methods that include grouping are the verbal-argumentative approach described by Giegrich and Schmitz (1996), as further developed by Schmitz and Paulini (1999), and the ranking method by Volkwein et al. (1996).

7.3. Weighting

Weighting (also sometimes referred to as "valuation" in some LCA circles) refers to using numerical factors based on value choices to facilitate comparison across impact category indicators (or normalised results). Weighting is often applied in the form of linear weighting factors:

$$\mathrm{EI} = \sum V_k N_k \text{ or } \mathrm{EI} = \sum V_k S_k \tag{4}$$

where EI is the overall environmental impact indicator, V_k is the weighting factor for impact category k, N is the normalised indicator and S is the category indicator from the characterisation phase.

Weighting remains a controversial element of LCA, as in other assessments—mainly because weighting involves social, political and ethical value choices (Finnveden, 1997). Not only are there values involved when choosing weighting factors, but also when choosing which type of method to use, and even in the choice of whether to use a weighting method at all. However, all weighting methods include scientific aspects—not only from natural sciences, but also from social and behavioural sciences as well as from economics. For example, techniques, knowledge and theories developed within decision analysis and environmental economics can be used for weighting in LCIA (Hertwich and Hammitt, 2001a,b; Powell et al., 1997; Seppälä et al., 2002).

Methods for weighting can be classified in different ways (Finnveden et al., 2002):

(1) A distinction can be made between methods based on impact indicators defined early (at midpoints) or late (e.g. at endpoints or for areas of protection), in the impact chain, as described in Section 5.
(2) A second distinction is between three major groups of methods:
    - Monetisation (here used as an umbrella term for all methods which have a monetary measure involved in the weighting factors)
    - Panel (a group of methods where the relative importance of damages, impact categories or interventions is derived from a group of people through surveys)
    - Distance to target (where characterisation results are related to target levels)
(3) A third distinction exists between expressed preference methods and revealed preference methods. Panel methods, as well as some monetisation methods, are based on expressed preferences. People are asked their preferences (for example, willingness-to-pay). On the other hand, some monetisation methods are based on revealed preferences. These monetised weighting factors are derived from reactions to different situations of individuals and/or organisations, such as insurance payouts, health care expenditures, fines, costs incurred for environmental cleanups and ecotaxes. Hofstetter and Müller-Wenk (2003) provided an overview of monetisation in the context of LCA and human health.

Table 2 illustrates the characteristics of some of LCIA methods that include predefined weighting approaches. This table distinguishes between the depth of modelling (to midpoints yielding indicators such as GWPs versus to indicators at the level of AoPs) and the type of subsequent weighting approach. Baumann and Rydberg (1994), Lindfors et al. (1995), Bengtsson and Steen (2000) and Dreyer et al. (2003) provided details of the differences between the results when some of the methods in Table 2 were applied in LCA studies. Finnveden et al. (2002) described and evaluated a number of these different weighting methods and approaches using a system of criteria, concluding that there is currently no method that fulfils all relevant criteria. But, Finnveden et al. recommended that

Table 2
Illustrative, non-exhaustive, summary of characteristic elements of some LCIA methodologies that include weighting approaches

Method[a]	Versions available[b]	Geographical scope[b]	Indicator basis (impact modelling depth)		Weighting basis		
			to midpoint	to AoP	Distance to target	Expert panel	Monetisation
Eco-scarcity	1991, 1997	Switzerland	partly		X		
EPS	1992, 1996, 2000	World		X			X
Ecoindicator	1995, 1999	Europe/Netherlands		X		X	
EDIP	1997, 2003	Denmark	X		X		
LIME	2003	Japan		X			X

[a] The references considered for the respective methods are: Eco-scarcity (BUWAL, 1998), EPS (Steen, 1999a,b), Ecoindicator (Goedkoop and Spriensma, 1999), EDIP (Wenzel et al., 1997) and LIME (Itsubo and Inaba, 2003).

[b] The original geographic scope of the listed methods. Midpoint methods (Ecoindicator-95, Eco-scarcity, EDIP), applying distance to target weighting, have been adapted to several countries and regions (e.g. Baumann and Rydberg, 1994; Hansen, 1999; Lee, 1999; Itsubo, 2000; Doka, 2002).

distance-to-target methods should not be used as weighting methods, unless it is explicit that all targets are of equal importance. Distance-to-target methods should be regarded as a type of normalisation technique where the targets are reference values.

8. Conclusions

LCA is a tool for comparing goods and services (products) and for identifying opportunities for reducing the impacts attributable to associated wastes, emissions and resource consumption. This paper provides a review of the LCIA phase. The paper outlines differences in the underlying modelling options and methodologies that currently facilitate calculation of impact indicators from life cycle inventory data.

A large number of models are available for calculating characterisation factors for use by LCA practitioners. Characterisation factors linearly express the relationship between inventory data and impact category indicators. Depending on the approaches selected, available indicators can provide estimates of the marginal changes to cumulative risks and potential impacts attributable to different product options. These indicators can be compared across impact categories, like climate change and toxicological effects, at an area of protection level such as human health.

The appropriate impact models, methodologies and application guidance for practitioners has been, or is being, defined through research and consensus building activities at the national and international scale. Nevertheless, as with any form of impact assessment tool, the science behind LCIA is continually evolving. Research and development activities include:

- Improving the depth and breadth of modelling environmental mechanisms in terms of consistency, comparability and providing a clearer identification of the environmental relevance of the indicator results in terms of marginal risks and the potential impacts to better facilitate integrated decision making.
- Identifying appropriate levels of spatial and temporal differentiation, including guidelines for when this differentiation is necessary to arrive at a decision.
- Quantifying the overall uncertainties of indicator results, establishing how to account for such uncertainties in the decision making process and identifying areas where modelling improvements would be beneficial.
- Further development of the natural science, social science and monetised techniques that facilitate comparison across impact categories (global warming, toxicological effects, etc.) and across areas of protection (resource consumption, human health, impacts on ecosystems, etc.) to better support integrated decision-making.

Acknowledgements

Five earlier forums addressing issues in LCIA and their associated summaries provided valuable inputs when writing this review—by SETAC (Udo de Haes, 1996; Barnthouse et al., 1997; Udo de Haes et al., 2002) and by the US EPA/UNEP/CML (Bare et al., 2002a,b, 1999). We also thank the unknown journal reviewers for their precise recommendations that facilitated improvements.

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